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Tracking coral loss in the Spermonde Archipelago of Indonesia: 32 years of satellite monitoring from 1990 to 2022

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ABSTRACT

An alarming amount of damage has occurred to coral reefs in Indonesia, with the causes of such damage either from human activities (anthropogenic causes) or natural processes. This study investigated the long-term dynamic changes in coral reef habitats in eight selected islands in the Spermonde archipelago, Indonesia, using remote sensing and historical information. A series of Landsat images recorded in different years (i.e. 1990, 1996, 2002, 2008, 2013, 2016, 2018, 2019, and 2022) were used to map the benthic habitats of each island (i.e. live coral, dead coral, seagrass, sand, rubble, mix bottom). The analysis estimated a total of 303.47 hectares or 76.26% loss of live coral in studied islands since 1990. In contrast, there was a substantial increase in dead coral algae and rubble, estimated at around 108.63 and 101.91 hectares, respectively. Strong and similar negative coefficient determination between live coral cover with both dead coral algae ($R^2 = 0.96$, Y = -2.95) and rubble ($R^2 = 0.94$, Y = -2.85) suggested that human activities (e.g. destructive fishing activities, damage from anchors, coral mining) and natural processes (e.g. sedimentation, coral bleaching, coral diseases, and coral predation) played comparable roles in coral reef degradation in Spermonde Archipelago. Furthermore, this study revealed that destructive fishing occurred on the island regardless it was inhabited or uninhabited and was influenced by the extent of its live coral cover. This study overall highlighted the ongoing degradation and factors contributing to coral loss in the Spermonde archipelago. Further research and monitoring efforts are necessary to understand the mechanisms driving the decline in live coral and the factors contributing to increases in dead coral algae and rubble. This knowledge will aid in developing targeted conservation strategies and management plans to protect the coral reef ecosystem in the Spermonde Archipelago and other affected regions in Indonesia.

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1. Introduction

Coral reefs are critical coastal ecosystems that provide valuable environmental products and services for the lives of coastal communities. However, this ecosystem is one of the most at risk marine ecosystems because of anthropogenic activities and the disruption of natural changes on both local and global scales (Lasagna et al. 2014). Their existence currently faces threats both from natural processes and human activities. Significant degradation of coral reefs has been noted in recent decades associated with destructive fishing (Edinger et al. 1998; Jompa 1996), sewage pollution (Tomascik, Suharsono, and Mah 1994), coral bleaching attributed to climate change (Arnold and Glynn 1991; Brown 1997; Hughes et al. 2017; Yusuf and Jompa 2012), dredging activities (Cunning et al. 2019; Luter et al. 2021), hurricanes and increased concentrations of atmospheric greenhouse gases that cause the global warming (Ampou et al. 2018). Coral reefs are very sensitive to temperature increases, and an increase in water temperature of 2°C can cause mass reef death (DeCarlo et al. 2017). It has also been found Burke et al., (2002) that human activities are the greatest cause of reef damage (greater than global warming and natural disasters).

Indonesia, with its extensive 95,181 km coastline, boasts a vast expanse of coral reefs covering approximately 50,000 km^2. These reefs are renowned for their remarkable biodiversity and intricate ecological complexity. Notably, Indonesia is part of the Coral Triangle, a global epicentre of tropical marine biodiversity encompassing six countries: Indonesia, Malaysia, the Philippines, Timor Leste, Papua New Guinea, and the Solomon Islands. The Spermonde Archipelago stands as one of the regions included within this ecologically significant Coral Triangle. The Spermonde Archipelago is a group of small islands located in the Liukang Tupabiring District, situated in the Sulawesi Strait between Sulawesi Island and Kalimantan Island in South Sulawesi Province, Indonesia. It consists of 54 small islands, of which 14 of them are not inhabited, and the other 130 are categorized as *gusung*, flat or lumpy areas of sand or coral fragments and shells which some are submerged only during low tide (Nurdin et al. 2014). The Spermonde Archipelago is rich in marine biodiversity and widely considered the largest coral reef fishery in Indonesia (Pet-Soede and Erdmann 1998).

Due to its very high coral reef biodiversity, Spermonde Archipelago contributes significantly to fishery activities in South Sulawesi. The majority of the people who inhabit the islands of the Spermonde Archipelago depend heavily on its marine resources therefore fishing is one of their main occupations. The increasing population in this area directly influences the utilization of marine space for fisheries, driven by the growing demand for fish consumption (Pet-Soede and Erdmann 1998) or to support their family's economy. However, it also has been known that destructive fishing activities occurred in this region for many years that destroying most of its coral reef ecosystems (Edinger et al. 1998; Jompa 1996; Pet-Soede and Erdmann 1998). Destructive fishing has been an ongoing issue, and a national concern as it also occurs in other islands throughout Indonesia.

Due to many natural and anthropogenic threats these days, it becomes more and more crucial to monitor and protect the coral reef's ecosystems. The satellite remote sensing approach is an instrument to monitor the coral reefs' condition and has been widely applied to map coastal habitat distributions around the globe. Mapping coral reefs can benefit from high-resolution satellite data. A detailed classification was made of more

than 150 habitat types using high-resolution (2 m) image data obtained by GeoEye-1 (Ampou et al. 2018). Studies of coral reef ecosystem habitat changes in the Red Sea, Hurghada, Egypt, and for the first time on Bunaken Island, North Sulawesi, Indonesia (El-Askary et al. 2014), have results for supporting coral reef areas management. Shallow water mapping using remote sensing technology is cost and time-efficient, and it enables extensive area cover when identifying coral reefs' condition (Hedley et al. 2016). In addition, satellite imagery can classify shallow water habitats based on spectral characteristics to map spatial and temporal distributions of coral reefs and seagrass ecosystems (Vahtmäe and Kutser 2013).

Utilizing satellite imagery data allows for the monitoring of past and current coral reef conditions (El-Askary et al. 2014). Landsat has been used for data acquisition on coral reefs since 1984, therefore have the advantage of a long-time series archive (Hedley et al. 2018). This study assesses the long-term dynamic changes of coral reef habitats in the Spermonde Archipelago, Indonesia from 1990 to 2022 through the utilization of Landsat imageries. An additional point of this study was the technique employed for retrospective analysis of images without ground truth data (i.e. 2008, 2002, 1996, 1990). In addition to remote sensing analysis, this study also investigated the correlation between the inhibition of the islands and the level of coral destruction. While the study focused on Spermonde archipelago as a case study, the issue of coral reef degradation extends to other islands across Indonesia. This study highlighted the challenges faced by coral reefs in the Spermonde, serving as a representative example for Indonesia. The methodologies and findings in this study can inform national-level conservation strategies, aiding in the preservation and management of coral reef ecosystems across the country.

2. Material and methods

2.1. Study sites

The Spermonde archipelago is located in the Makassar Strait, situated off the southwestern coast of the island of Sulawesi, specifically in the province of South Sulawesi. The archipelago encompasses a large area and consists of numerous small islands, islets, and coral reefs. The Spermonde archipelago is known for its rich marine biodiversity, including coral reefs, seagrass beds, and diverse marine life. This study focused on eight islands in the archipelago which were divided into two categories: small inhabited islands and small uninhabited islands. The small inhabited islands consisted of Badi, Bonetambung, Karangrang, Ballang Lompo, Ballang Caddi, and Gondongbali meanwhile the small uninhabited islands were Samatellu Pedda and Kondingareng Keke (Figure 1). The presence of few inhabitants in Samatellu Pedda was only observed after 2016, however, since the studied year started from 1990, Samatellu Pedda was therefore considered an uninhabited island in this study.

2.2. Multitemporal data

Multitemporal Landsat satellite images were utilized to investigate coral distribution changes on each island for 32 years from 1990 to 2022 (Table 1). The images were obtained freely from the USGS website (http://earthexplorer.usgs.gov/). All images were

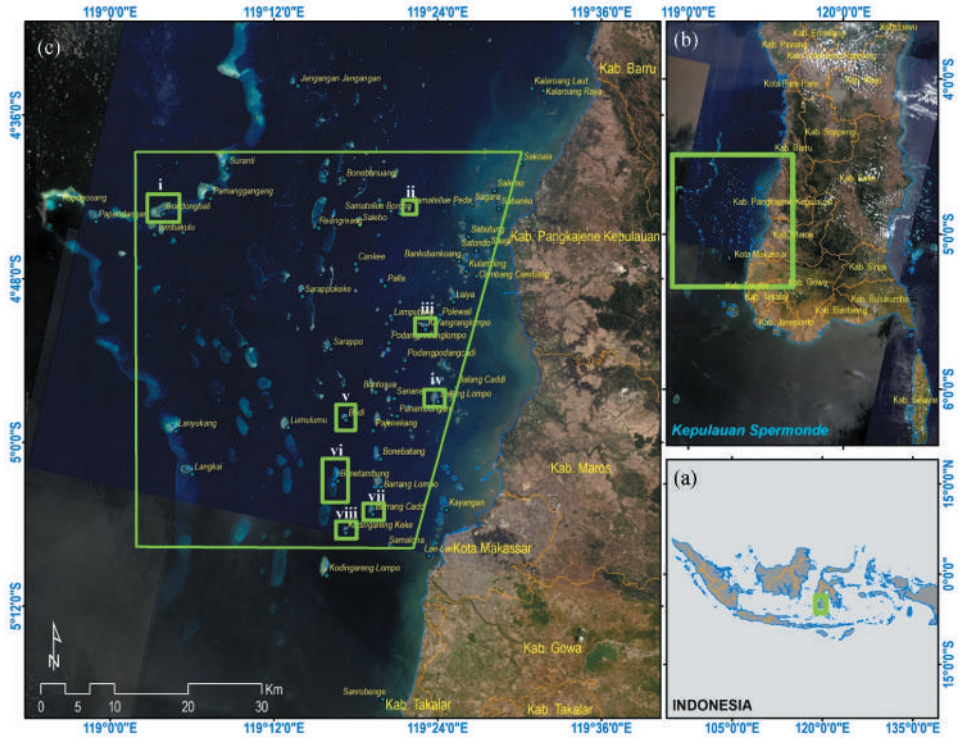


Figure 1. Map showing the Spermonde archipelago in the context of (*a*) Indonesia and (*b*) South Sulawesi province. (*c*) eight Spermonde islands selected as the study area were: (i) Gondong Bali, (ii) Sametellu Pedda, (iii) Karanrang, (iv) Balang Lompo, (v) Badi, (vi) Bonetambung, (vii) Barang Caddi and (viii) Kodingareng Lompo.

Table 1. Acquisition date for the Landsat images.

Acquisition date	Used for	Sensor	Spatial resolution (m)
2022 – April – 12	all island	Landsat 9 OLI TIRS	15
2019 – January – 06	all except Kodingareng Keke		
2018 – April – 25	Kodingareng Keke	Landsat 8 OLI TIRS	
2016 – September – 10	all except Gondong Bali		
2013 – October – 04	Gondong Bali		
2008 – August – 19	all island	Landsat 7 ETM	30
2008 – August – 3	all island		
2002 – June – 24	all island	Landsat 5 TM	
1996 – April – 28	all island		
1990 – December – 16	all island	Landsat 4 TM	

obtained from scene path/row number 114/64, all cloud-free, and projected to UTM 50s at reference datum WGS 84. The spatial resolution of the images used was 30 metres for images from 1996 and 1990 and 15 metres for images from 2002, 2008, 2013, 2016, 2018, 2019, and 2022. Two image acquisitions were used for the year 2008 due to the presence of scan line errors in the images, therefore they were required special treatment for image correction (see section Scan line correction). Apart from that, image acquisition from 2013 and 2018 was only used for Gondong Bali and Kodingareng Keke, respectively, to fit ground truth data availability.

2.3. Image processing

In a given order image processing performed in this study consisted of scan-line error correction, geometric correction, atmospheric correction, pan-sharpening, water column corrections, unsupervised classification, and re-classification. Following unsupervised classification, two different reclassification procedures were performed between years with ground truth data (*i.e.*, 2013, 2016, 2018, 2019, 2022) and years with no ground truth data (*i.e.*, 2008, 2002, 1996, 1990). The overall workflow is presented in the diagram in Figure 2.

2.3.1. Scan-line error correction

Scan line error correction on the 2008 images was carried out using the image fusion technique. The image fusion process involved aligning and registering multiple images taken at different times to eliminate the scan line errors and performed on remote sensing software. In this study, Landsat images recorded on August 3rd and 19th, 2008, were fused to obtain clear images of the study location for the year 2008.

2.3.2. Geometric correction

The geometric correction used in this study was coordinate transformation (geometric transformation). It was performed using the second-order Polynomial method, with a minimum of 6 (six) Ground Control Points (GCPs). The images from 2022 and 2019 were corrected by directly obtaining GCPs in the field. Both images were then used as a reference for the other years' images. The second-order polynomial equation used is as follows (Mohammed, Eisa, and Elhaj 2013):

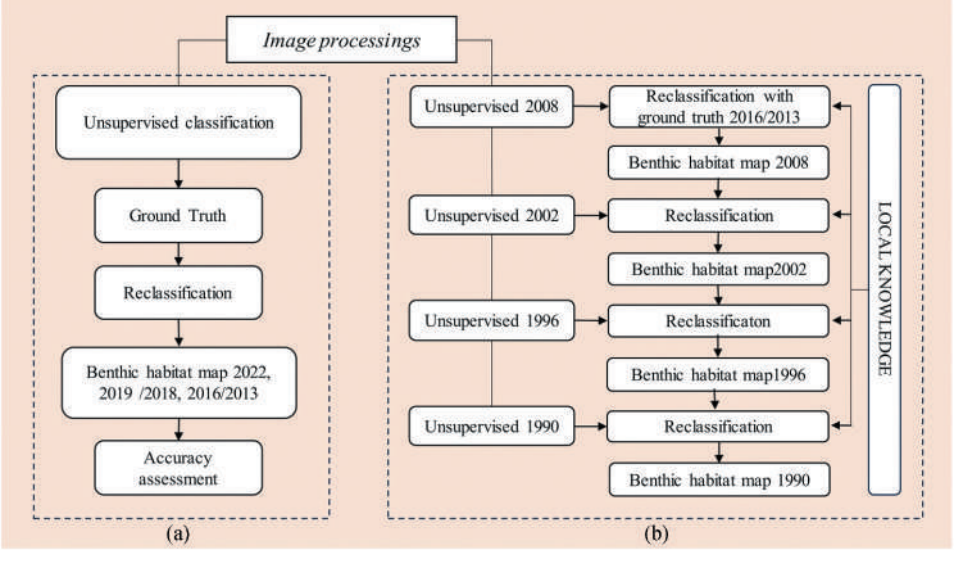


Figure 2. Workflow diagram benthic habitat maps classification process followed in this study. Following the image processing, part (*a*) shows steps performed for years with ground truth data, and part (*b*) for years with ground truth data.

$$P(x,y) = a_1 + a_2x + a_3y + a_4x^2 + a_5xy + a_6y^2 \quad (1)$$

Where p (x, y) represents the rectified coordinate (output), and x and y are the original coordinates (input), while a1, a2, a3, a4, a5, and a6 are the parameters of the second-order Polynomial transformation.

2.3.3. Atmospheric correction

The atmospheric correction process was performed using Dark Object Subtraction (DOS) module. DOS module removed the haze effects that modify the spectral signature and reduce classification accuracy (Chavez 1988). It derives the corrected DN (Digital Number) values solely from the digital data with no outside information. The haze DN value was directly selected from the DN frequency histogram of the Landsat images.

2.3.4. Pan-sharpening

In this study pan-sharpening was applied to images from 2022, 2019, 2018, 2008, and 2002, to increase the spatial resolution of the images from 30 to 15 metres and to calculate coral conditions in greater detail (Hanaizumi and Akiba 2008). Pan-sharpening is the process of merging a higher spatial resolution panchromatic image with a lower spatial resolution multispectral image to produce a higher spatial resolution multispectral image (Rahaman, Hassan, and Ahmed 2017).

2.3.5. Water column correction

Water column correction is necessary when mapping coastal habitats in shallow waters since the water column attenuates light (Green 2000; Lyzenga 1978; Stumpf et al. 2003). The Depth Invariant Index (DII) algorithm for water column correction combined the blue (490 nm) and green (560 nm) bands of visible light and was applied to enhance the contrast between the land and water areas of the study location. The DII algorithm is known to be effective in reducing the effects of water column attenuation, which can be a major challenge in mapping shallow coastal habitats (Eugenio et al. 2017; Hedley et al. 2018; Siregar et al. 2019). The DII is given by the following equation:

$$\mathrm{DII} = \log(\mathrm{xi}) + [(\mathrm{Ki}/\mathrm{Kj})^* \log(\mathrm{xj}) \quad (2)$$

where *xi*, *xj*, and *ki/kj* are blue band and green band values and the ratio of the attenuation coefficient of bands © and *j* is shown with © for the blue band and *j*.

2.3.6. Unsupervised classification

The ISODATA (Iterative Self-Organizing Data Analysis Technique) algorithm (Dhodhi et al. 1999) was used on the images to produce unsupervised classification images. The ISODATA classification algorithm resulted in 20 clusters where pixels with almost the same spectral value were as one class (Li, Liu, and Huang 2020) and each cluster was assigned to a particular benthic class category with the help of in situ data (reclassification).

2.4. Ground truth and reclassification

The primary objective of ground truth surveys was to validate and verify the correspondence between the real world-objects and their respective classes on the unsupervised

maps. Ground truth surveys were conducted three times in 2016 (1st ground truth), 2019 (2nd ground truth), and 2022 (3rd ground truth). However, for Gondong Bali, the 1st ground truth data used was collected in 2013 and the 2nd ground truth data used for Kodingareng Keke was collected in 2018. The number of ground truth points collected at each study site varied based on the characteristics and the cover of the shallow water area, ranging from 41 to 110 points per island. The date of the survey and the number of ground truth points for each island can be seen in Tables 2, 3, and Table 4.

Reclassification was performed on unsupervised images based on the ground truth data. The classes having the same object based on the ground truth data were merged into one class. The reclassification process resulted in the assignment of the initial 20 classes into six distinct benthic habitats categories *i.e.*, sand, rubble, dead coral algae

Table 2. 1st ground truth statistics and overall accuracy of classified images for all islands in 2022.

		3rd ground truth			
No	Island	Date	Number of points	Number accuracy test points	Overall accuracy (%)
1	Godongbali	12 April 2022	70	98	70
2	Samatellu Peda	22 July 2022	43	61	74
3	Karanrang	13 April 2022	79	83	73
4	Ballang Lompo	13 April 2022	80	97	76
5	Badi	22 July 2022	60	64	78
6	Bonetambung	8 June 2022	57	78	78
7	Barrang Caddi	8 June 2022	70	99	75
8	Kodingareng Keke	8 June 2022	50	66	73

Table 3. 2nd ground truth statistics and overall accuracy of classified images for all islands in 2019 (2018 for Kodingareng Keke island).

		2nd ground truth			
No	Island	Date	Number of points	Number of accuracy test points	Overall accuracy (%)
1	Godongbali	9 June 2019	*51*	141	75
2	Samatellu Peda	9 June 2019	*42*	40	77
3	Karanrang	11 June 2019	*98*	135	75
4	Ballang Lompo	11 June 2019	*70*	100	84
5	Badi	10 June 2019	*55*	48	81
6	Bonetambung	10 June 2019	*60*	62	82
7	Barrang Caddi	13 July 2019	*80*	123	78
8	Kodingareng Keke	13 July 2018	*55*	57	82

Table 4. 3rd ground truth statistics and overall accuracy of classified images for all islands in 2016 (2013 for Gondongbali island).

		1st ground truth			
No	Island	Date	Number of points	Number of accuracy test points	Overall accuracy (%)
1	Godongbali	11 June 2013	107	224	84
2	Samatellu Peda	26 July 2016	41	57	79
3	Karanrang	27 July 2016	94	150	77
4	Ballang Lompo	27 July 2016	110	193	84
5	Badi	26 July 2016	50	49	78
6	Bonetambung	24 July 2016	45	53	83
7	Barrang Caddi	24 July 2016	60	79	79
8	Kodingareng Keke	24 July 2016	59	44	80

(DCA), live coral, seagrass, and mix bottom (mix between rubble, sand, and seagrass). Reclassification with ground truth data as a validator ensured that the assigned classes correctly represented the six actual real-world benthic habitat categories.

2.5. Examination of retrospective image analysis

The technique used to classify the images without ground truth data (i.e. 2008, 2002, 1996, 1990) involved using a validated map from the nearest available year as a proxy to classify the earlier images (Figure 3). To assess the accuracy of this technique, initially, a reclassification test was performed on 2016 images of each island (2013 for Gondong Bali island) – in which their ground truth data were available (1st ground truth). Firstly, the 2013 and 2016 unsupervised images of each island were reclassified using the 2nd ground truth of each island, as detailed in Table 3 and illustrated in Figure 4 and Appendix B. Then, these reclassified images were compared against the 1st ground truth data for the accuracy assessment. The accuracy obtained using this approach revealed a classification of accuracy level of more than 75% (Appendix A). Green et al., (2000) stated that the accuracy level that met the threshold of 60% or higher is significant enough to demonstrate the image classification accuracy. Figure 3 shows the benthic maps of each island obtained from this technique.

Based on the satisfactory accuracy achieved from the initial test (>75%), the same approach was then applied to images from 2008, 2002, 1996, and 1990 (see the workflow diagram Figure 2(b). The unsupervised images from 2008 were reclassified using the 1st ground truth data, which was the closest available ground truth data to the year 2008. Then, these reclassified results of the 2008 images were subsequently used to reclassify the unsupervised image from 2002, and so on. After the reclassification, a refinement process was carried out in several locations to address anomalous results. For example, there were instances where live coral classes were also identified near the coastline, which

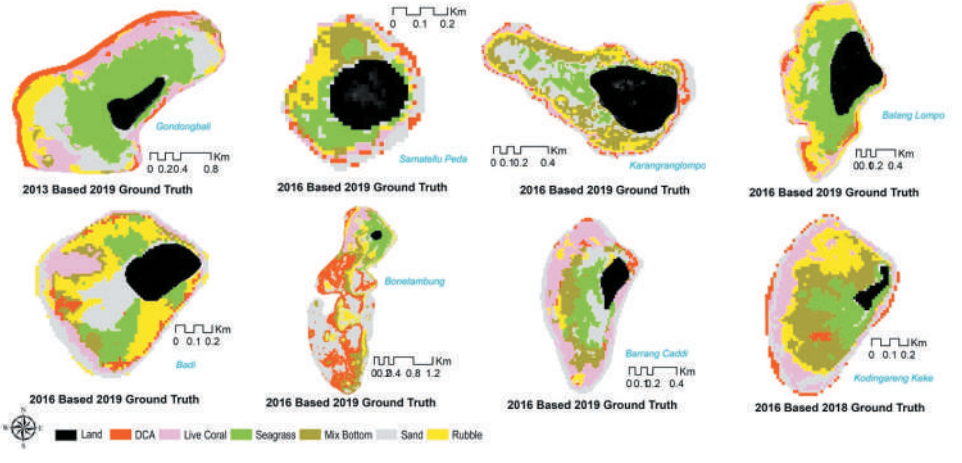


Figure 3. Illustration of the 2016 benthic habitat maps of the study location (2013 for Gondongbali) which were reclassified using the 2nd ground truth data from 2019 (2018 for Kodingareng Keke). The accuracy level of these images were tested against the 1st ground truth data and showed an accuracy level of more than 75% each.

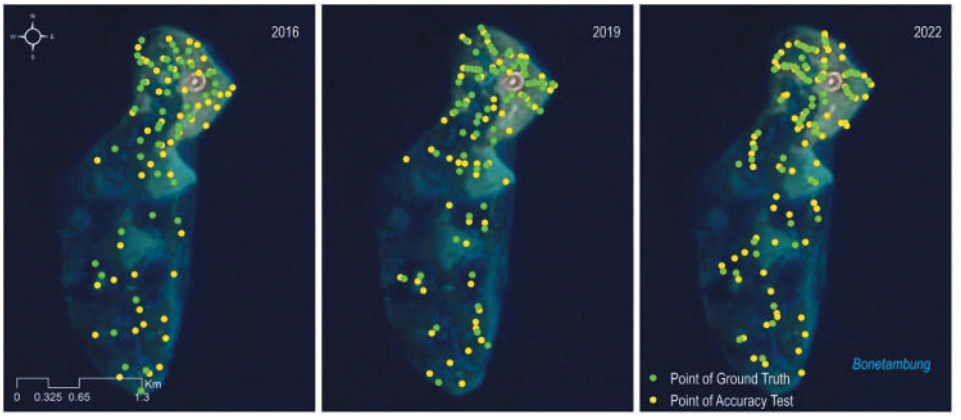


Figure 4. Examples of a set of ground truth (green) and accuracy test (yellow) points on Bonetambung island for the years 2016, 2019, and 2022.

contradicts the expected habitat for coral. This may have occurred due to the object-based approach used in the unsupervised classification, which grouped objects with similar spectral reflections into a single class (Island et al. 2005). In this particular case, the area classified as coral near the coastline could potentially be seagrass instead of coral. To support the validation, local knowledge was integrated. A map from respected years was shown to fishermen and incorporated their knowledge. Fishermen's expertise and familiarity with the local ecosystem could help identify potential misclassifications and improve the accuracy of the classification (Lauer and Aswani 2008).

3. Interviews and field observations

Interviews were conducted using a purposive sampling technique. Respondents were selected based on the criteria that they were fishermen actively engaged in the profession, aged above 50 years, and have a minimum of 20 years of experience as fishermen, which would make them knowledgeable about historical coral reef locations and conditions. The selected respondents also represented individuals residing on the north, south, west, and east sides of the island with 8–10 respondents for each side. Specific questions were asked including but not limited to how they conduct their fishing activities, if they have used coral reef for construction or building material, and the type of fishing gear used. Additionally, on-island observations were carried out to photograph the utilization of coral reefs by the inhabitants.

3.1. Accuracy test

A confusion matrix was used to assess the accuracy of the reclassified images against the ground truth data (Rosenfield and Fitzpatrick-Lins 1986). Confusion matrices were only tested on images that had ground truth data (images from 2013, 2016, 2018, 2019, 2022). The number of points for the accuracy test points varied between 44 to 224 points (see Tables 2, Table 3, Table 4). These accuracy test points were strategically chosen to represent various areas and habitats of each island. They were carefully distributed across

the island to ensure a representative sample for accuracy (see Table 2, 3, and 4). Figure 4 provides examples of ground truth and accuracy test points carried out in Bonetambung Island for the years 2016, 2019, and 2022 (see Appendix B for the complete figure of other islands).

4. Results and discussion

4.1. Accuracy assessments of classified images

A comparative evaluation of the reclassified images versus independent ground truth points for the year in which the ground truth data available and all studied locations showed an overall accuracy ranging between 70–84%. As shown in Appendix C, the results of the reclassification were displayed as column data, representing the producer's accuracy, while the row data represented the ground truth data were used to calculate the user's accuracy. The expected overall accuracy value must meet the threshold of 60% or higher (Green et al. 2000), which is significant enough to demonstrate the image classification accuracy. The classification results of 2022 validated with 3rd ground truth exhibited accuracy levels ranging between 70–78% (Table 2). Similarly, the results for 2019 (2018 for Karanrang Island) validated with 2nd ground truth demonstrated accuracy levels of 75–84% (Table 3). Lastly, the images from 2016 (2013 for Gondongbali island) validated with 1st ground truth data showed accuracy levels between 77–84% (Table 4). Overall, the accuracy assessments of the classified images affirm the reliability of the classification technique employed in this study. These findings indicate that the reclassified maps were well aligned with the ground truth data for the respective years. The accuracy tests provided a measure of confidence in the classification methodology and suggested that the benthic habitat classification maps can be relied upon for further analysis and decision-making processes.

4.2. Spermonde benthic habitats dynamic through remote sensing

Benthic habitat classification plays a crucial role in understanding the dynamics of marine ecosystems. In this study, Landsat images were used to generate benthic habitat classification maps of the studied locations into six categories i.e. live coral, DCA, seagrass, mix bottom, rubble, and sand. Moreover, the multitemporal analysis illustrated the spatial variations in the distribution of the benthic habitat throughout the studied year. Figure 5 provides the benthic habitat maps from 1990 to 2022 of Ballang Lompo, Bonetambung, and Gondongbali (see Appendix D for the complete benthic habitat maps of all islands). Below each map, the zoomed-in version showcased the spatial distribution changes of the benthic habitat in specific locations on the island. It is worth noting that since 1990, the live coral has transitioned into either DCA, rubble, sand, or mix bottom category.

The application of remote sensing provided a valuable means of assessing the alterations of live coral within the study area. By utilizing these classified maps, it became possible to quantify in hectares (ha) the coverage changes of each benthic habitat category. This approach offered crucial insights into the dynamics of the Spermonde benthic habitats for 32 years. In total, the dynamic cover of the benthic habitat category from all the studied locations, as depicted in Figure 6, shows that in 1990 live coral was the

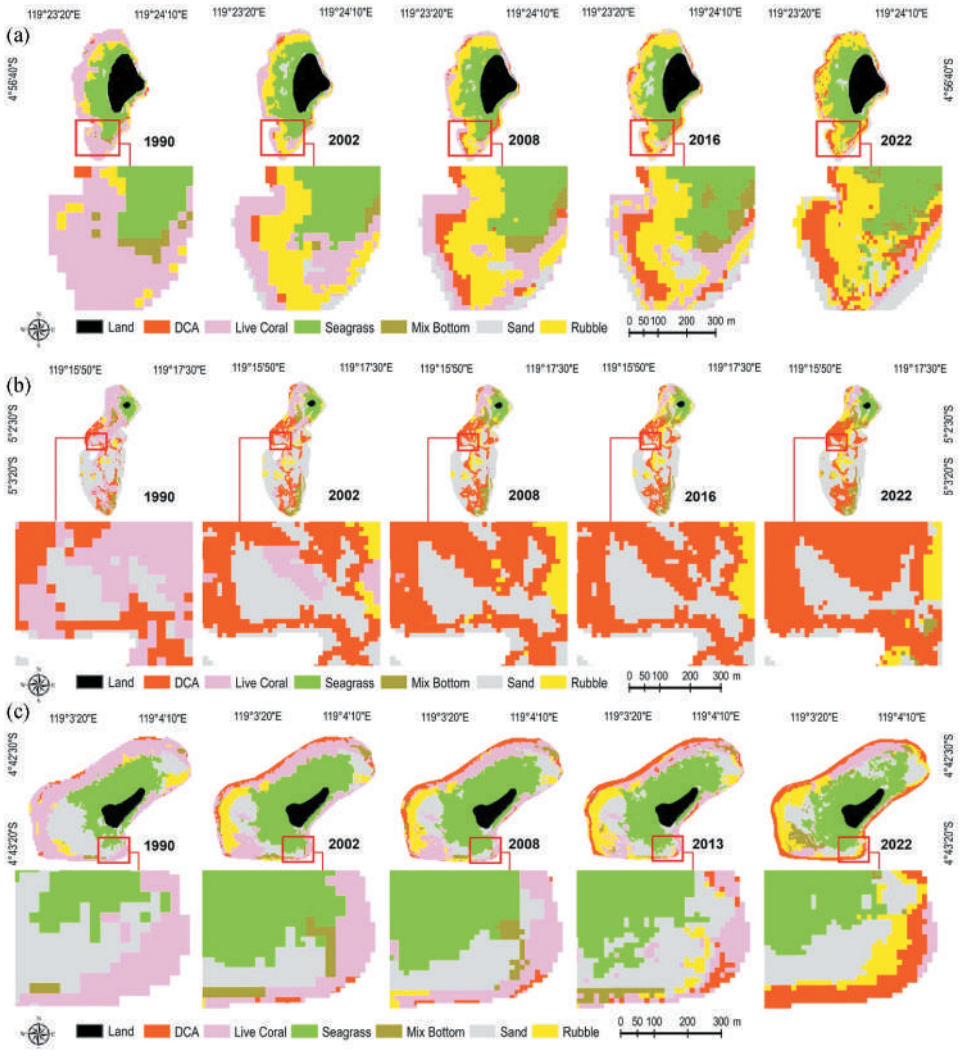


Figure 5. Benthic habitat maps of Ballang Lompo (*a*), Bonetambung (*b*), and Gondong Bali (*c*) from 1990 to 2022. The zoomed-in images offer a detailed view of specific locations, illustrating how the benthic habitat has changed throughout the studied period.

largest benthic habitat category, followed by sand and seagrass. However, since 1990, the live coral has shown a declining trend and eventually fell below the sand category after 1996. This decline persisted, resulting in live coral becoming the benthic habitat with the smallest cover area as of 2022. Meanwhile, DCA and rubble emerged as the third and fourth largest benthic habitat categories in Spermonde. Previous studies have indicated that reef degradation in the Spermonde Archipelago has been strongly linked to anthropogenic mechanical damage (e.g. destructive fishing activities, coral mining, anchor damage) and land-based pollution (e.g. sewage, sedimentation and industrial pollution) (Edinger et al. 1998; Nurdin et al. 2015), and coral bleaching events attributed to climate change (Yusuf and Jompa 2012). Due to the different image acquisition years used for

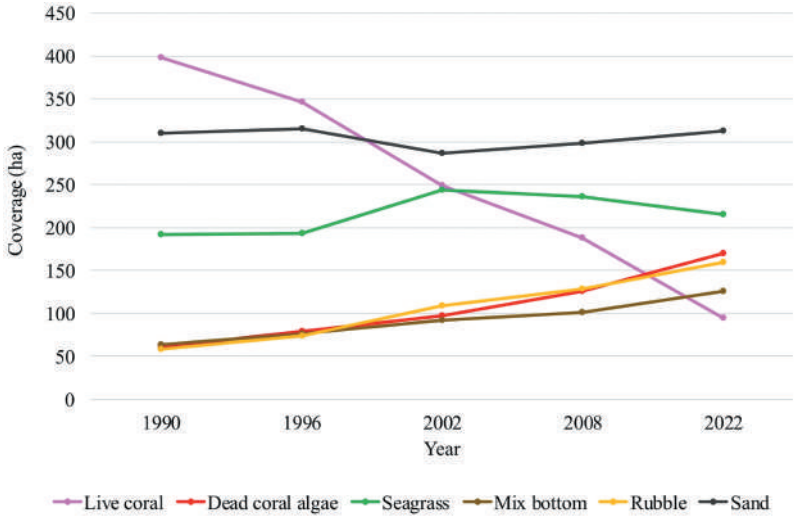


Figure 6. The Spermonde benthic habitat category dynamics measured in hectares (ha) derived from Landsat image classifications from 1990 to 2022. Live coral experienced a significant decline, whereas dead coral algae, rubble, and mix bottom categories showed a steady increase. By 2022, live coral had the smallest cover among other benthic habitat categories.

Gondong Bali (i.e. 2013) and Kodingareng Keke (i.e. 2018), the collective results for the years 2016 and 2019 were not included in Figure 6. This was to ensure that the comparison was based on data from the same time points.

In detail, Figure 7 revealed that in 1990, in each island – except Bonetambung – live coral had the largest cover area in comparison to other benthic habitat categories. Gondong Bali exhibited the largest live coral cover with 128.04 ha, followed by Bonetambung (115.99 ha), Ballang Lompo (46.17 ha), Barrang Caddi (36.62 ha), Badi Island (23.62 ha), and Karanrang (21.24 ha). The two uninhabited islands, Kodingareng Keke and Samatellu Pedda, exhibited the smallest live coral cover among the studied locations, with 20.43 ha and 5.82 ha, respectively. However, the graphs also show that there was a significant decrease in live coral cover on each island and an increase in rubble, DCA, mixed bottom, and sand cover. Bonetambung and Gondong Bali experienced the highest increase in DCA since 1990, with an increase of 45.37 ha and 40.29 ha, respectively. Conversely, Bonetambung experienced the most significant decrease in live coral cover, with a loss of 101.17 ha. Gondong Bali, despite remaining as the island with the largest live coral cover (44.32 ha) as of 2022, came second for the live coral loss with 87.69 ha since 1990.

Bonetambung, Karanrang, and Gondong Bali experienced a significant increase in the mix bottom category, while the other islands showed a stable but not significant increase (except for Barrang Caddi, which showed a slight decrease). There was an overall increase in sand cover observed across all islands, with significant increases seen in the inhabited islands of Badi and Ballang Lompo, as well as the uninhabited islands of Kodingareng Keke and Samatellu Pedda. The seagrass category exhibited fluctuating patterns, with no significant changes observed over the years. Considering all the islands studied in this

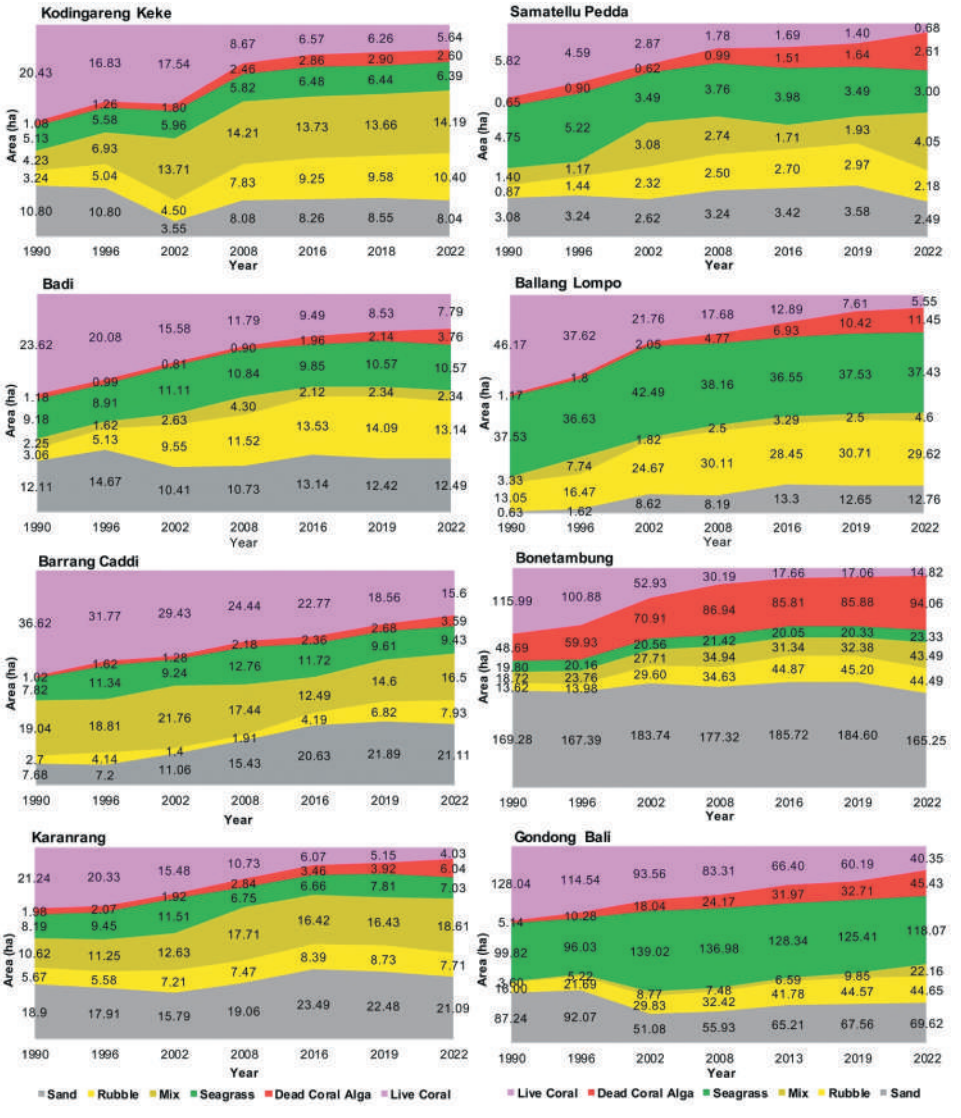


Figure 7. Benthic habitat cover changes in hectares (ha) derived from Landsat image classifications from 1990 to 2022 on each studied island.

research, the mix bottom category demonstrated a fluctuating pattern but surpassed the live coral category in 2022. Additionally, sand remained the benthic habitat category with the largest cover area after 1996, followed by seagrass.

4.3. Correlation of benthic habitat dynamics

A coefficient determination analysis was conducted to examine the potential dynamic changes correlation between each benthic habitat category. The coefficient of determination (R^2) and the slope coefficient (Y) values as shown in Figure 8, can be interpreted as the proportion of variation in the dependent variable that can be explained by the

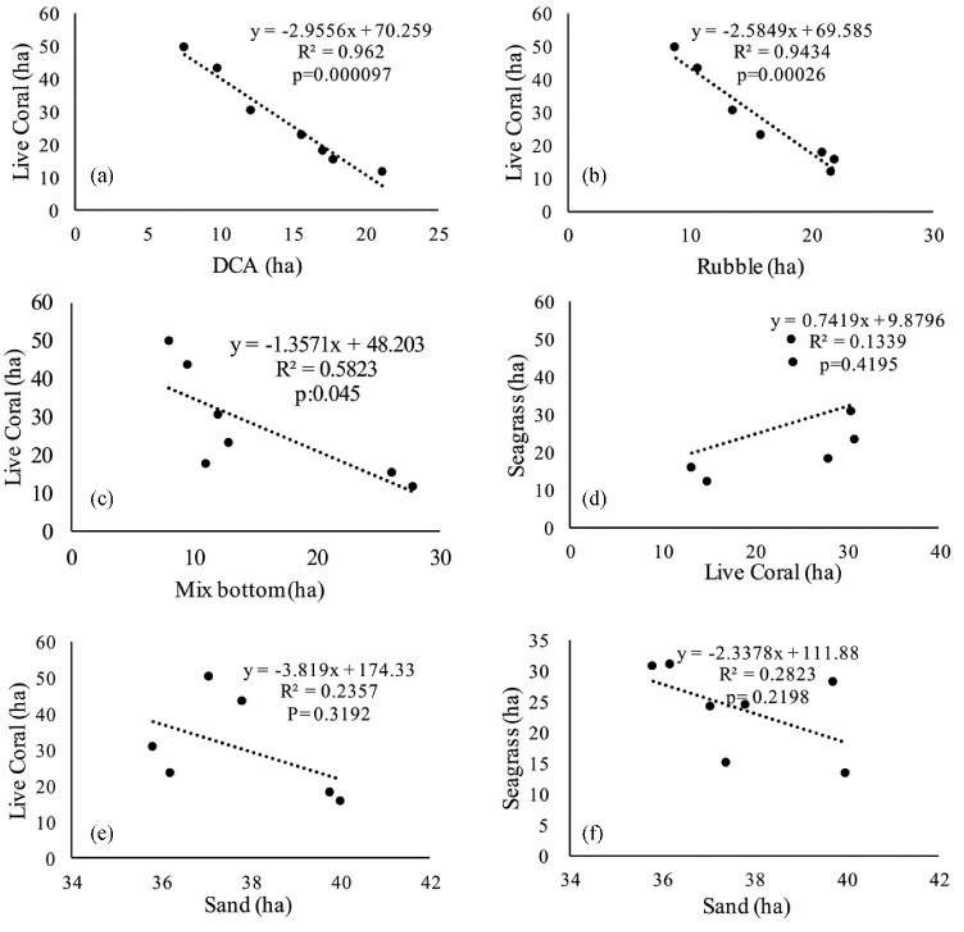


Figure 8. Coefficient determination analysis between benthic habitat categories. A strong negative correlation was revealed between live coral with dead coral alga (DCA) (*a*) and between live coral with rubble (*b*). A moderate correlation between live coral and mix bottom (*c*). Low correlation between live coral with seagrass (*d*), sand with live coral (*e*), and sand with seagrass (*f*).

variation in the independent variables. Therefore, in the context of the relationship between DCA and live coral cover (Figure 8(a)), and between rubble and live coral cover (Figure 8(b)) the high R^2 values and negative Y coefficient suggest a strong negative correlation. The R^2 value of 0.962 for DCA suggests that approximately 96.2% of the variation in live coral cover can be explained by the variation in DCA cover. The Y value of −2.955 for DCA suggests that, on average, for each unit increase in DCA cover, there was an estimated decrease of 2.955 units in live coral cover. Similarly, the R^2 value of 0.94 for rubble suggests that around 94% of the variation in live coral cover also can be explained by the variation in rubble cover. The Y value of −2.85 for rubble indicates there was an estimated decrease of 2.85 units in live coral cover for each unit increase in rubble cover.

The strong negative coefficient determination between DCA and rubble cover with the decrease in live coral cover suggests that both human activities and natural processes might have similar detrimental effects on coral reefs in the Spermonde. Human activities,

such as the use of bombing, coral mining and anchor damage, can physically break down coral structures, resulting in the formation of rubble. Land-based pollution, including sedimentation from coastal development, can lead to increased sediment deposition on corals, covering them and hindering their growth and survival. Additionally, coral bleaching, often associated with climate change, can result in the loss of live coral tissue and subsequent colonization by algae.

Coefficient analysis between mix bottom and live coral (Figure 8(c)) with the R^2 value of 0.58 and a Y value of −1.357 suggest a moderate negative correlation indicating that as the mix bottom cover increased, the live coral cover tended to decrease. However, between seagrass and live coral (Figure 8 (d)), live coral and sand (Figure 8(e)), and between sand and seagrass (Figure 8(f)) all show weak correlation with an R^2 value of 0.133, 0.235, and 0.282, respectively. Seagrass and coral tend to occupy distinct ecological niches within the marine environment, where seagrass typically thrives in shallow coastal areas with sandy substrates and coral reefs are often found in deeper waters with hard substrates, thus leading to limited interactions between them. Therefore, seagrass may have some influence on reducing sand cover in Spermonde, however, it was not very significant. Factors such as anthropogenic mechanical damage and sedimentation might contribute to the accumulation of sand, but their impact on the increase sand cover was not substantial.

4.4. Population growth and coral reef degradation in spermonde

The inhabitation of the Spermonde archipelago is closely linked to the availability of marine resources, particularly the coral reefs. The indigenous Bugis people have a long tradition of seafaring and fishing, and many settlements were established on the islands to take advantage of the abundant marine resources. Over time, these settlements grew and developed into larger communities, with more people attracted to the area by the prospect of a sustainable livelihood provided by the sea. Today, the Spermonde archipelago is home to more than ten thousand people, with a diverse mix of ethnic and religious groups, including Bugis and Makassar. While most of the islands have become densely populated (Figure 9 (a) – (d), (f)) the others remain uninhabited (Figure 9 (g) and Figure 9(h)). The small islands of Spermonde have become increasingly vulnerable to various challenges due to population growth, increased competition for marine resource utilization, and rapid urbanization (Idrus, Badawing, and Irwanto 2021). The consequent increase in human settlements also has resulted in intensified land-based pollution, leading to further deterioration of the marine ecosystem.

The remote sensing analysis conducted in this study provides evidence of a significant decrease in live coral cover in the Spermonde archipelago since 1990. Table 5 shows a total decline of live coral approximately 303.47 hectares or 76.26% from 1990 to 2022. The islands exhibited varied percentages of live coral loss, ranging from 57.40% to 88.32%. The summary from Table 6, highlighted the increased variation of dead coral algae and rubble across different islands. Bonetambung and Gondong Bali stood out with the highest increase in DCA and rubble cover, which is attributable to the fact that these islands initially had a higher live coral cover compared to others. This result aligns with the insights gathered from the conducted interviews, where it was revealed that people tend



Figure 9. UAV photographs obtained in 2018 and 2019 of the six inhabited islands: (*a*) Barrang Caddi, (b) Karanrang, (*c*) Ballang Lompo, (*d*) Bonetambung, (*e*) Badi, (f) Gondongbali, and two uninhabited islands: (*g*) Kodingareng Keke, and (*h*) Samatellu Pedda.

Table 5. Summary of live coral loss cover in Spermonde derived from remote sensing analysis.

	Cover (ha)			
Island	1990	2022	Cover of loss (ha)	Percentage of loss (%)
Uninhabited				
Kodingareng Keke	20.43	5.64	14.79	72.39
Samatellu Pedda	5.82	0.68	5.14	88.32
Inhabited				
Badi	23.62	7.79	15.83	67.02
Ballang Lompo	46.17	5.55	40.62	87.98
Barrang caddi	36.62	15.6	21.02	57.40
Bonetambung	115.99	14.82	101.17	87.22
Karanrang	21.24	4.03	17.21	81.03
Gondong bali	128.04	40.35	87.69	68.49
Total	**397.93**	**94.46**	**303.47**	**76.26**

to engage in fishing activities in areas with still more abundant live coral, as these areas would still harbour rich marine biodiversity.

Considering the initial live coral cover of each island in 1900, Samatellu Pedda experienced the highest loss, with 88,32% of its live coral cover depleted. In contrast, Barrang Caddi had the lowest loss, with only 57.40%. These findings suggest that even

Table 6. Summary of increases of dead coral algae and rubble cover in Spermonde derived from remote sensing analysis (in hectares).

	Dead coral algae (ha)			Rubble (ha)		
Island	1990	2022	Increase	1990	2022	Increase
Kodingareng Keke	1.08	2.60	1.52	3.24	10.40	7.16
Samatellu Pedda	0.65	2.61	1.96	0.87	2.18	1.31
Badi	1.18	3.76	2.58	3.06	13.14	10.08
Ballang Lompo	1.17	11.45	10.28	13.05	29.62	16.57
Barrang caddi	1.02	3.59	2.57	2.70	7.93	5.23
Bonetambung	48.69	94.06	45.37	13.62	44.49	30.87
Karanrang	1.98	6.04	4.06	5.67	7.71	2.04
Gondong bali	5.14	45.43	40.29	16.00	44.65	28.65
Total	**60.91**	**169.54**	**108.63**	**58.21**	**160.12**	**101.91**

uninhabited islands could suffer relatively higher levels of coral loss compared to inhabited ones. Furthermore, Table 6 highlights the substantial increase in rubble cover observed in Kodingareng Keke, which surpassed the quantity found in Barang Caddi and Karanrang. These results confirm the insights gathered from interviews conducted with the local people in the Spermonde archipelago, which revealed that destructive fishing practices also occur in uninhabited islands.

4.5. Contributing factors to the decline of live coral in spermonde

The cause of coral damage can be grouped into either human activities (anthropogenic causes) or natural processes. Coral damage caused by human activities includes excessive fishing, extracting biota or coral reefs, sedimentation, and activities that are not environment-friendly. Damage caused by natural processes is divided into damage caused by physical processes such as waves by a strong low pressure and damage caused by biological processes such as an explosive increase in crown-of-thorns starfish population (coral predation), coral bleaching, and coral disease. Sedimentation occurs when excessive sediments settle on corals, resulting in poor water quality, reducing light penetration, and hindering photosynthesis, ultimately leading to the death of coral and an increase in algae (Arias-González et al. 2017; Rogers and Ramos-Scharrón 2022). Coral bleaching, often triggered by environmental stressors like increased water temperatures (DeCarlo et al. 2017), results in the expulsion of symbiotic algae, causing the loss of colour and reduced energy production in corals (Baker, Glynn, and Riegl 2008). Diseases can also spread among corals, weakening their defences and leading to the decline of coral cover (De et al. 2012; Muller et al. 2012). Predators such as crown-of-thorns starfish and coral-eating snails can further exacerbate the conversion of live coral to dead coral (Caballes and Rivera-Posada 2014). On the other hand, destructive fishing activities, including the use of bombs, damage from anchors, and coral mining significantly contributed to the creation of rubble (Nurdin et al. 2015). These activities physically break apart the coral, causing fragmentation and the accumulation of rubble. The use of explosives and harmful fishing practices can destroy coral colonies, while storms and anchors can dislodge and damage corals, leading to the formation of rubble.

The study revealed that the overall increase of DCA in Spermonde was higher, although not significant, compared to rubble with respective increased areas of 108.63 and 101.91 hectares. Islands such as Samatellu Pedda, Bonetambung, Karanrang, and Gondong Bali experienced higher increases in DCA might suggest that natural processes

such as coral bleaching, disease, and predation were more influential factors in coral degradation in these locations. Coral reef bleaching events have led to mass mortality in numerous reefs worldwide over the last four decades (Arnold and Glynn 1991; Brown 1997; Bruno, Selig, and Freckleton 2007; Edmunds 1994; Oliver and Oliver 1999). In Spermonde, a four-year monitoring programme (2007–2010) of coral reefs found that coral bleaching occurred in 2009–2010, resulting in a 12.45% decrease in live coral cover in Spermonde (Yusuf and Jompa 2012). Six coral diseases including white syndromes and black band disease were recorded in Spermonde in 2011 (Muller et al. 2012). Moreover, Plass-johnson et al., (2015) investigated that the *Acanthaster* outbreak in Spermonde has caused the loss of half of the live coral cover in Bonetambung and Barrang Lompo.

Islands such as Kodingareng Keke, Badi, Ballang Lompo, and Barrang Caddi showed higher increases in rubble compared to DCA. This suggests that human activities such as destructive fishing activities (e.g. use of bombs), damage from anchors, and coral mining might have a greater impact on coral destruction in these islands. Destructive fishing methods are still prevalent in the Spermonde region (Nurdin et al. 2015). With the increasing population of the islands in the Spermonde Archipelago, the demand for additional settlement units also rises. Many island residents excavated sand and stony coral for building materials, which has significantly impacted the coral reef ecosystem (Figure 10). Interviews conducted in Gondongbali, Samatellu Pedda, Badi, and Ballang Lompo revealed more than 60% of respondents use coral reefs for construction materials for their houses (Appendix E).

The decline of live coral in Spermonde can also be attributed to the growing demand for ornamental fisheries, which has resulted in unsustainable harvesting practices and extensive damage to coral reefs. The Spermonde archipelago is among the four major sites in Indonesia for the collection of ornamental coral, contributing significantly to the trade (Ferse et al. 2012). The extraction of ornamental fish from these reefs has become a significant source of income for local communities, leading to the overexploitation of coral reef ecosystems. This overharvesting, coupled with destructive fishing methods, has caused a significant decline in live coral cover and an increase in DCA, rubble, and sand.



Figure 10. The collected coral reefs used as construction materials in Spermonde for house foundations (*a*) and embankments (*b* - *d*) were photographed during the survey in 2019. This provides evidence that coral destruction has been occurring in the region for a significant period.

5. Conclusion

The remote sensing analysis conducted on the eight study locations in the Spermonde archipelago from 1990 to 2022 revealed a significant decline in live coral cover, with an approximate loss of 303.47 ha or 76.26%. In contrast, there was a substantial increase in dead coral and rubble cover, with an estimated increase of 108.63 and 101.91 ha, respectively. The variation in the increased levels of dead coral algae (DCA) and rubble across different islands suggests that certain factors such as human activities or natural processes might have a greater influence on specific locations. Islands such as Kodingareng Keke, Badi, Ballang Lompo, and Barrang Caddi experienced higher increases in rubble, which might suggest that human activities such as the use of bombs, coral mining, and damage from anchors have a more influence on the coral loss of these islands. On the other hand, Samatellu Pedda, Bonetambung, Karanrang, and Gondong Bali experienced higher increases in dead coral algae, which might suggest that natural processes such as coral bleaching, coral disease, and coral predation have a greater impact in these areas. However, collectively, strong and similar negative correlations between live coral and both dead coral algae ($R^2 = 0.96$, $Y = -2.95$) and between live coral and rubble ($R^2 = 0.94$, $Y = -2.85$) suggest that overall both human activities and natural processes played comparable roles in the degradation of coral reefs in Spermonde.

This study also revealed that destructive fishing practices also occurred on uninhabited islands and were driven by the existing live coral cover on the island. These findings suggested that there is an ongoing degradation of the coral reef ecosystem and emphasized the multi-faceted nature of reef decline and the interconnectedness between factors contributing to coral degradation in the Spermonde Archipelago. Continued research and monitoring are necessary to better understand the specific mechanisms driving the decline in live coral cover and the factors contributing to the increase in dead coral algae and rubble. This knowledge will inform targeted conservation strategies and management plans to safeguard the fragile coral reef ecosystem in the Spermonde Archipelago and beyond.

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No potential conflict of interest was reported by the author(s).

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Appendix A

Table A1. Accuracy test values (producer's, user's, and overall accuracy) for retrospective image classification analysis (2016 image).

	Producer accuracy						User accuracy						
Island	Live Coral	Dead Coral Algae	Rubble	Mix	Sand	Seagrass	Live Coral	Dead Coral Algae	Rubble	Mix	Sand	Seagrass	Overall accuracy (%)
Badi	78	86	67	63	100	67	70	75	75	83	70	86	76
Bonetambung	70	88	78	63	73	100	88	70	88	50	89	88	77
Kanranrang	83	87	68	81	88	93	77	83	89	79	88	88	75
Ballang Lompo	89	93	71	62	79	94	80	87	81	82	79	81	81
Barang Caddi	76	64	64	90	89	91	83	70	88	69	57	91	76
Gondong Bali	90	79	61	46	84	96	56	92	82	92	59	86	75
Samatellu Pedda	64	71	73	67	89	90	78	100	67	55	80	90	75
Kodingareng Keke	89	60	73	71	80	100	80	100	89	63	67	88	80

Appendix B

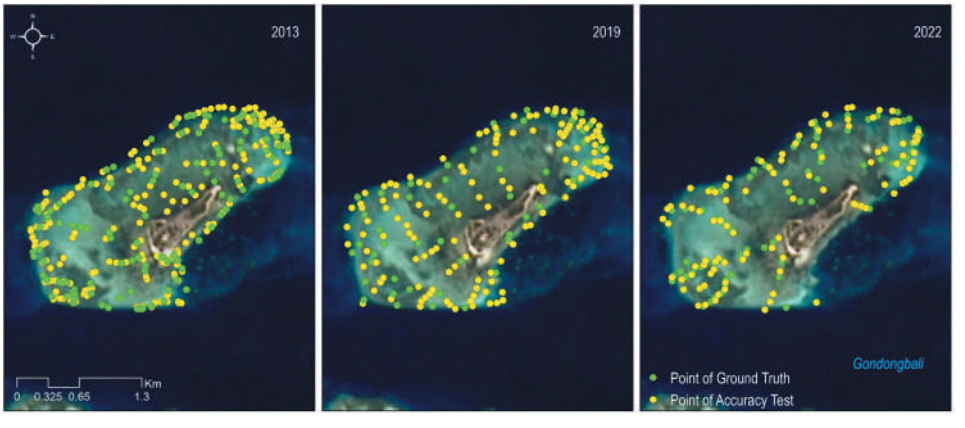


Figure B1. Illustration of a set of points used for ground truth (green) and accuracy test (yellow) analysis for Gondongbali island in 2013, 2019, and 2022.

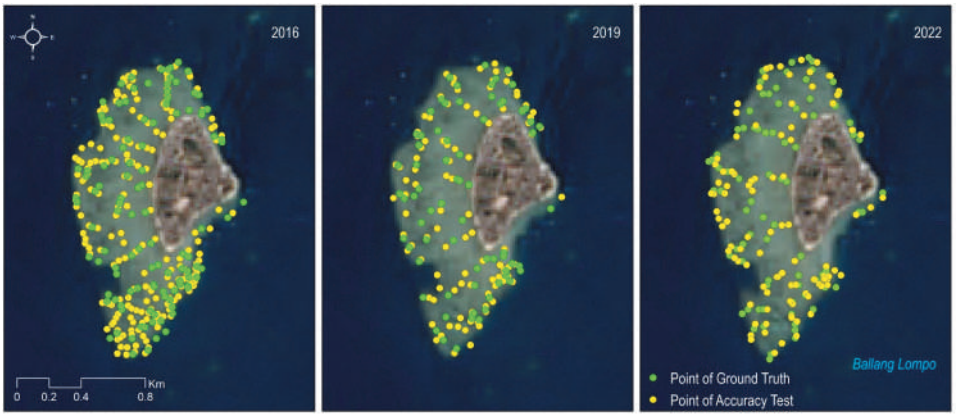


Figure B2. Illustration of a set of points used for ground truth (green) and accuracy test (yellow) analysis for Ballang Lompo island in 2016, 2019, and 2022.

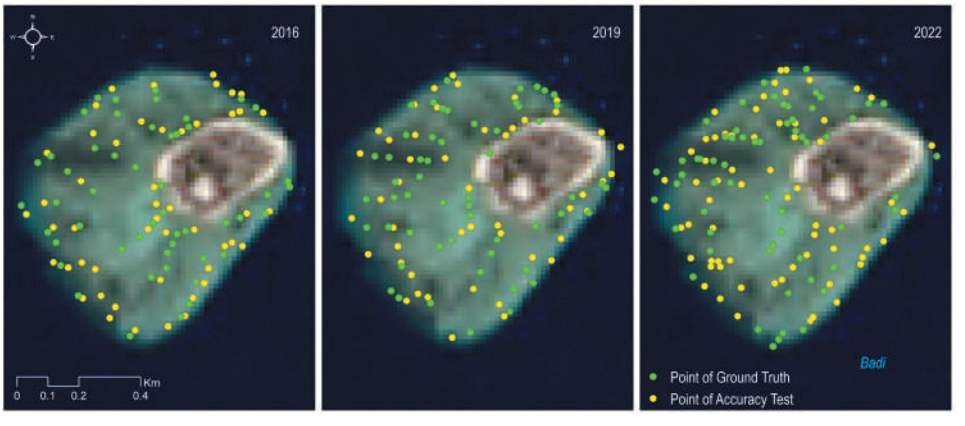


Figure B3. Illustration of a set of points used for ground truth (green) and accuracy test (yellow) analysis for Badi island in 2016, 2019, and 2022.

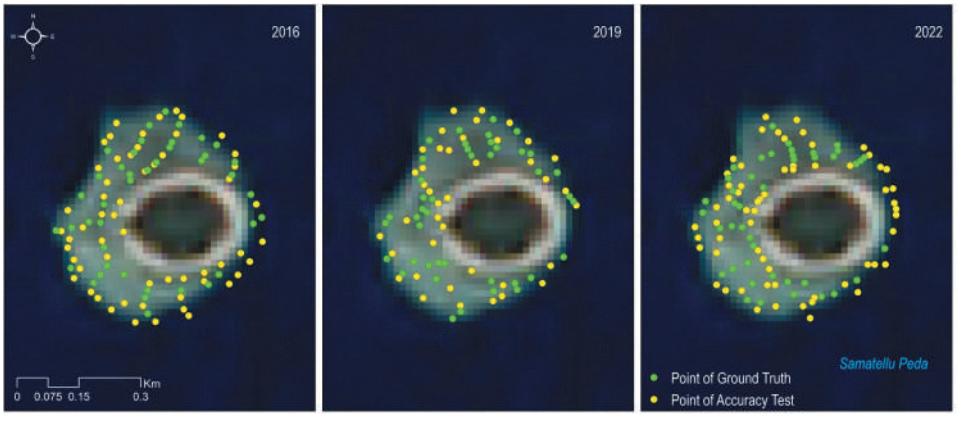


Figure B4. Illustration of a set of points used for ground truth (green) and accuracy test (yellow) analysis for Samatellu Peda island in 2016, 2019, and 2022.

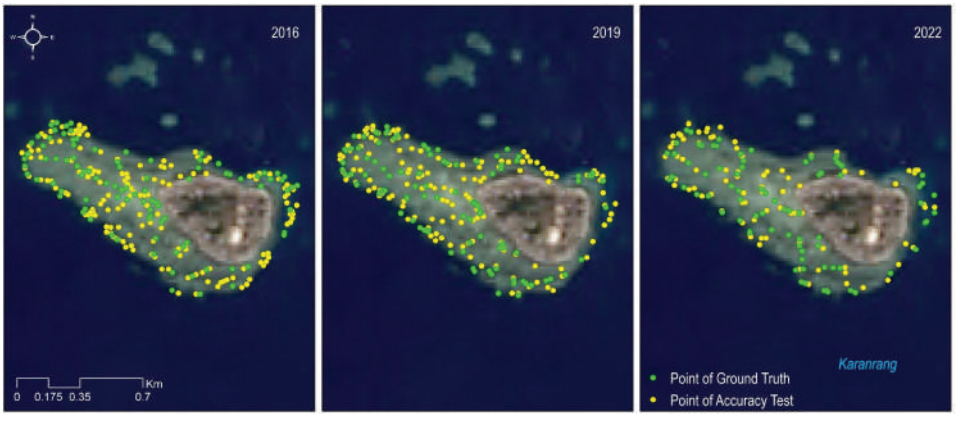


Figure B5. Illustration of a set of points used for ground truth (green) and accuracy test (yellow) analysis for Karanrang island in 2016, 2019, and 2022.

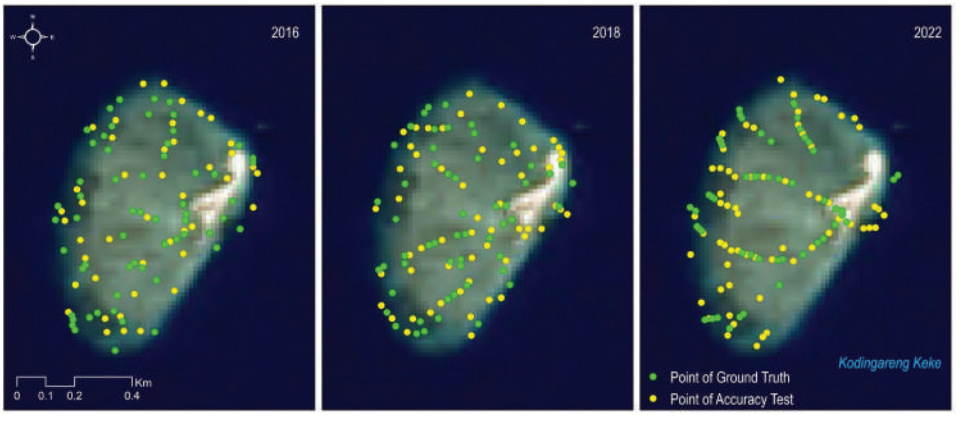


Figure B6. Illustration of a set of points used for ground truth (green) and accuracy test (yellow) analysis for Kodingareng Keke island in 2016, 2018, and 2022.

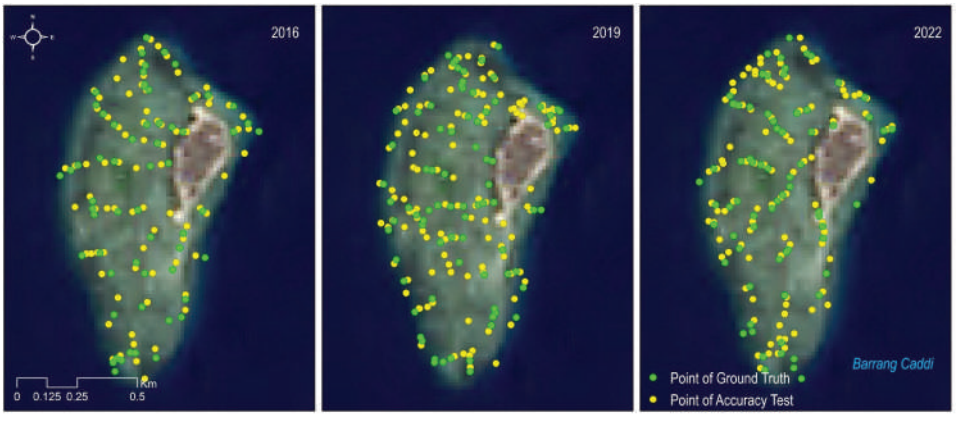


Figure B7. Illustration of a set of points used for ground truth (green) and accuracy test (yellow) analysis for Barrang Caddi in 2016, 2019, and 2022.

Appendix C

Table C1. Accuracy test values (producer's, user's, and overall accuracy) for the 2016 image classification in all islands (2013 for Gondongbali).

	Producer accuracy						User accuracy						
Island	Live Coral	Dead Coral Algae	Rubble	Mix	Sand	Seagrass	Live Coral	Dead Coral Algae	Rubble	Mix	Sand	Seagrass	Overall accuracy (%)
Badi	100	71	78	63	100	89	75	83	88	83	88	89	84
Bonetambung	70	100	89	63	73	86	100	73	73	63	89	86	79
Kanranrang	83	91	72	74	94	97	73	88	95	77	94	91	77
Ballang Lompo	86	93	76	69	89	94	81	81	86	80	89	88	84
Barang Caddi	80	73	64	67	89	100	77	80	78	80	80	79	78
Gondong Bali	95	83	71	72	89	92	80	83	79	82	86	92	83
Samatellu Pedda	82	71	64	67	89	100	75	83	88	60	89	83	79
Kodingareng Keke	89	60	73	71	80	100	80	100	89	63	67	88	80

Table C2. Accuracy test values (producer's, user's, and overall accuracy) for the 2019 image classification in all islands (2018 for Kodingareng Keke).

	Producer accuracy						User accuracy						
Island	Live Coral	Dead Coral Algae	Rubble	Mix	Sand	Seagrass	Live Coral	Dead Coral Algae	Rubble	Mix	Sand	Seagrass	Overall accuracy (%)
Badi	78	80	60	63	100	82	54	80	75	83	100	82	75
Bonetambung	70	70	73	78	90	83	88	54	73	70	90	100	77
Kanranrang	86	80	72	86	85	95	76	83	86	78	94	88	75
Ballang Lompo	94	88	74	76	92	87	79	82	89	81	86	87	84
Barang Caddi	83	88	80	78	78	90	83	100	86	70	88	56	81
Gondong Bali	90	82	65	65	95	96	72	79	75	81	90	92	82
Samatellu Pedda	100	63	57	67	100	100	67	71	100	75	80	86	78
Kodingareng Keke	86	100	60	77	100	100	86	75	82	83	82	88	82

Table C3. Accuracy test values (producer's, user's, and overall accuracy) for the 2022 image classification in all islands.

	Producer accuracy						User accuracy						
Island	Live Coral	Dead Coral Algae	Rubble	Mix	Sand	Seagrass	Live Coral	Dead Coral Algae	Rubble	Mix	Sand	Seagrass	Overall accuracy (%)
Badi	89	54	47	64	100	100	57	70	70	70	100	75	70
Bonetambung	80	83	62	60	91	89	57	71	87	82	71	80	74
Kanranrang	75	83	75	73	91	93	69	79	92	73	91	88	73
Ballang Lompo	87	91	61	64	100	80	68	63	88	74	92	80	76
Barang Caddi	91	77	63	76	89	82	69	67	89	81	89	82	78
Gondong Bali	86	79	67	56	100	86	71	79	71	77	94	75	78
Samatellu Pedda	90	64	69	58	100	89	64	70	90	70	86	80	75
Kodingareng Keke	75	83	75	73	91	93	69	79	92	73	91	88	73

Appendix D

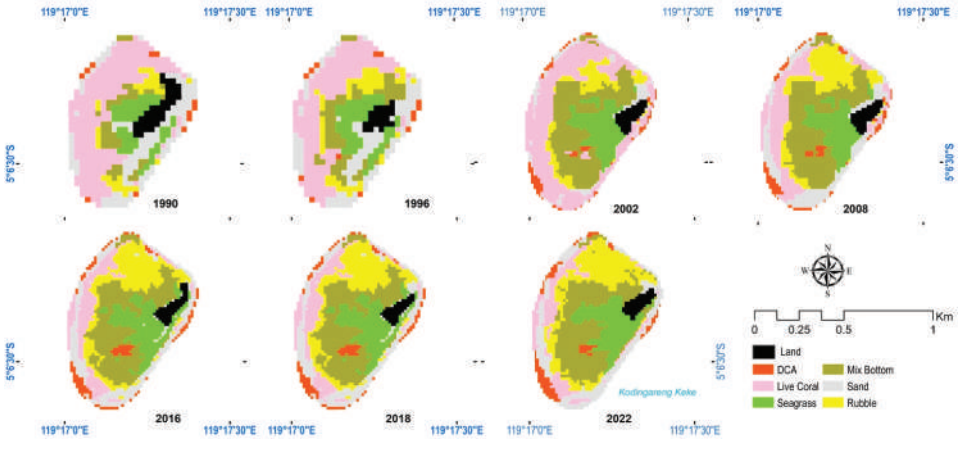


Figure D1. Benthic habitat maps of Kodingareng Keke from 1990–2022 derived from Landsat images. Images from 1990 and 1996 have 30 metres spatial resolution, and images from 2002 to 2022 have 15 metres spatial resolution.

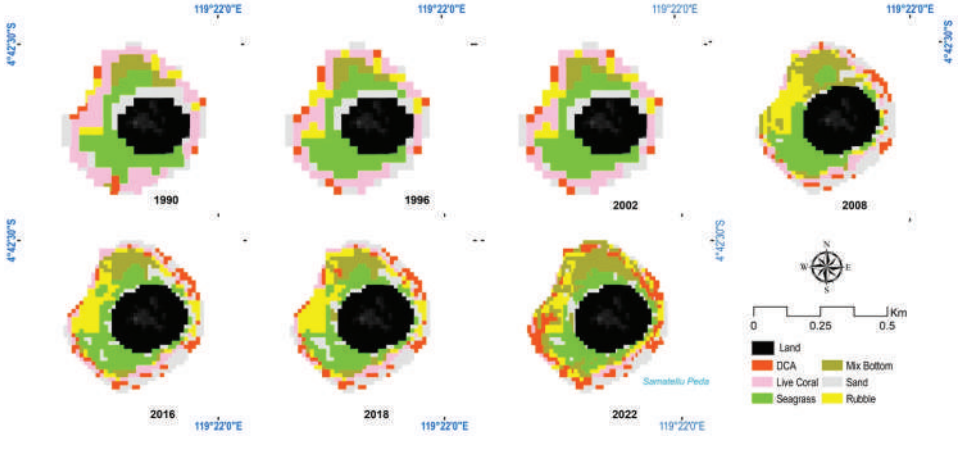


Figure D2. Benthic habitat maps of Samatellu Peda island from 1990–2022 derived from Landsat images. Images from 1990 and 1996 have 30 metres spatial resolution, and images from 2002 to 2022 have 15 metres spatial resolution.

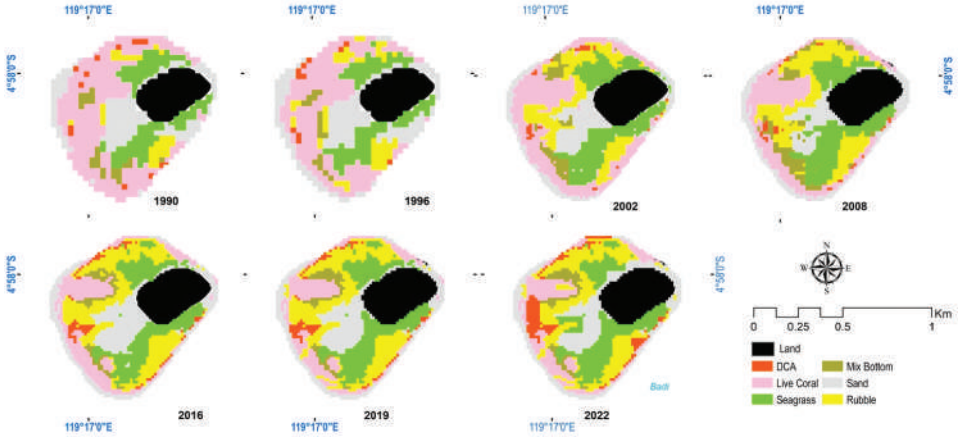


Figure D3. Benthic habitat maps of Badi island from 1990–2022 derived from Landsat images. Images from 1990 and 1996 have 30 metres spatial resolution, and images from 2002 to 2022 have 15 metres spatial resolution.

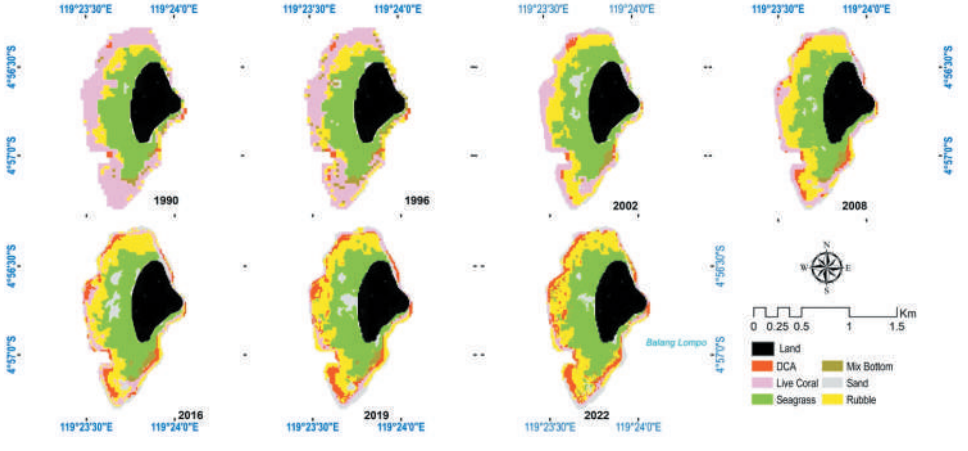


Figure D4. Benthic habitat maps of Ballang Lompo from 1990–2022 derived from Landsat images. Images from 1990 and 1996 have 30 metres spatial resolution, and images from 2002 to 2022 have 15 metres spatial resolution.

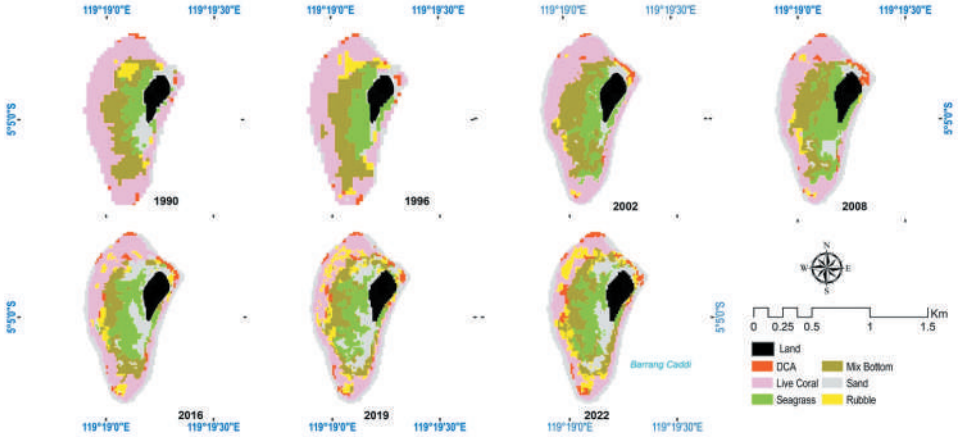


Figure D5. Benthic habitat maps of Barrang Caddi island from 1990–2022 derived from Landsat images. Images from 1990 and 1996 have 30 metres spatial resolution, and images from 2002 to 2022 have 15 metres spatial resolution.

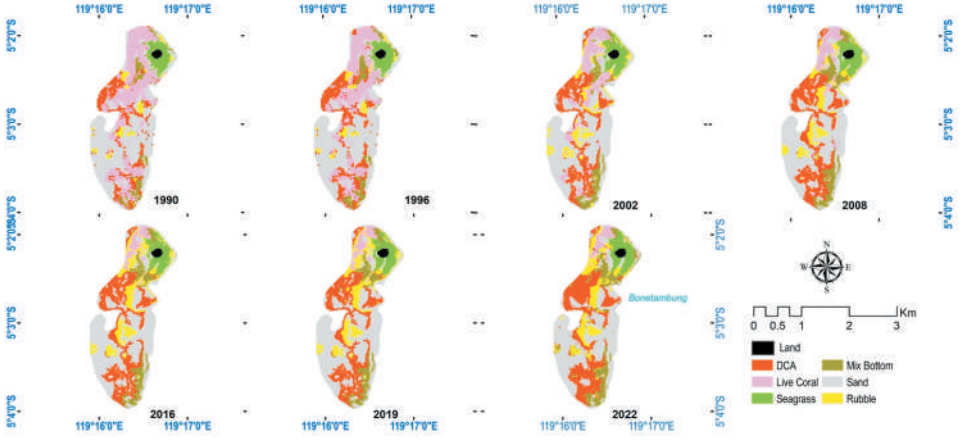


Figure D6. Benthic habitat maps of Bonetambung island from 1990–2022 derived from Landsat images. Images from 1990 and 1996 have 30 metres spatial resolution, and images from 2002 to 2022 have 15 metres spatial resolution.

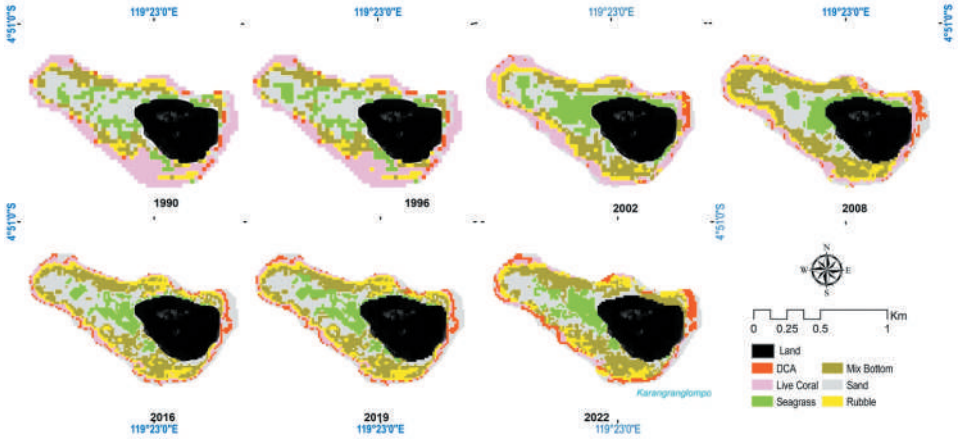


Figure D7. Benthic habitat maps of Karanrang island from 1990–2022 derived from Landsat images. Images from 1990 and 1996 have 30 metres spatial resolution, and images from 2002 to 2022 have 15 metres spatial resolution.

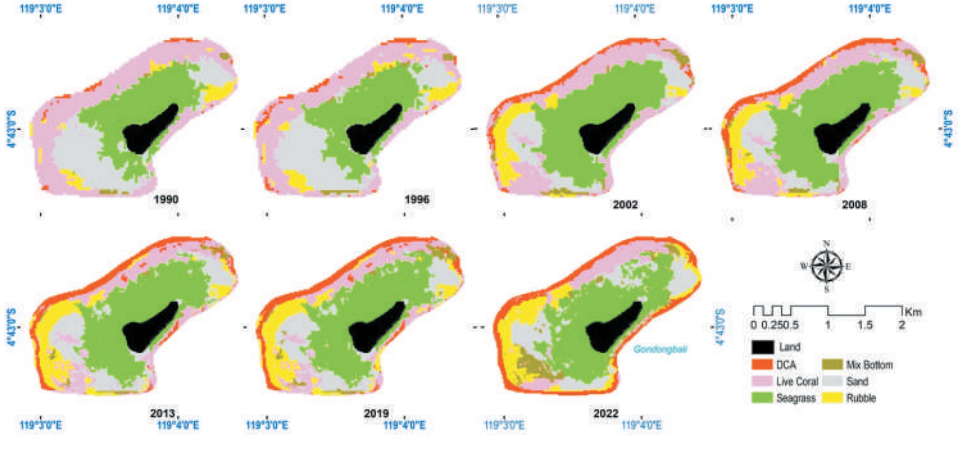


Figure D8. Benthic habitat maps of Gondong Bali island from 1990–2022 derived from Landsat images. Images from 1990 and 1996 have 30 metres spatial resolution, and images from 2002 to 2022 have 15 metres spatial resolution.

Appendix E

Table E1. The number of respondents on each island was Gondongbali island (34 people), Samatellu Pedda island (10 people), Badi island (33 people), Ballang Lompo island (38 people), Ballang Caddi island (40 people).

		Activity Type		
No	Name of Islands	Use of coral reef as a building material	The installation of Bubu/Rakkang	Aquaculture
1	Gondongbali	82%	-	15%
3	Samatellu Lompo	100%	-	-
4	Samatellu Pedda	100%	-	-
5	Samatellu Borong	100%	100%	-
6	Pala	100%	-	-
7	Lamputang	100%	-	-
8	Badi	61%	-	-
9	Lae-Lae	32%	28%	-
10	Ballang Lompo	95%	37%	100%
11	Ballang Caddi	90%	18%	70%
12	Pajenekang	63%	-	88%
13	Bontosua	68%	-	100%
14	Sanane	77%	-	40%